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LCA COMMUNICATION AND LCA FOR ISO LABELS

IMPACT 2002+, ReCiPe 2008 and ILCD's recommended practice for characterization modelling in life cycle impact assessment: a case study-based comparison

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Abstract

Purpose The European Commission has launched a recommended set of characterization models and factors for application in life cycle impact assessment (LCIA). However, it is not known how this recommended practice, referred to as the ILCD 2009, performs relative to some of the most frequently used alternative LCIA methodologies. Here, we compare the ILCD 2009 with IMPACT 2002+ and ReCiPe 2008, focusing on characterization at midpoint based on a case study comparing four window design options for use in a residential building.

Methods Ranking of the four window options was done for each impact category within each methodology. To allow comparison across the methodologies both in terms of total impact scores and contribution patterns for individual substances, impact scores were converted into common metrics for each impact category.

Results and discussion Apart from toxic impacts on human health and ecosystems, all studied methodologies consistently identify the same window option as having the lowest and the highest environmental impact. This is mainly because few processes, associated with production of heat, dominate the total impacts, and there is a large difference in demand for heat between the compared options. Despite this general agreement in ranking, differences in impact scores are above 3 orders of magnitude for human health impacts from ionizing radiation and ecosystem impacts from land use, and they lie between 1 and 3 orders of magnitude for metal depletion and for toxicity-related impact categories. The differences are somewhat smaller (within 1 order of magnitude) for the impact categories respiratory inorganics and photochemical ozone formation, and are within a factor of 3 for the remaining impact categories. The differences in impact scores in our case study are brought about by the differences in underlying characterization models and/or substance coverage, depending on the impact category.

Conclusions In spite of substantial differences in impact scores for the individual impact categories, we find that the studied LCIA methods point to the same conclusion with respect to identifying the alternative with the lowest environmental burden and ascribe this to the fact that few processes are driving the main environmental impacts, and there is large difference in demand for output from these processes between the compared options. Even though the overall conclusions remain the same for our case study, the choice of the ILCD's recommended practice over the existing alternatives does matter for the impact categories ionizing radiation and land use and all toxicity-related impact categories.

Keywords Characterization models · Comparison · Impact categories · Impact scores · Interpretation · Life cycle assessment

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1 Introduction

Differences in characterization models and their substance coverage for individual impact categories have earlier been identified as influential on the results of life cycle assessments (LCAs), sometimes able to change the conclusions of

comparative LCA studies and often leading to different ranking of substances in terms of major contributors to the environmental impact (e.g. Dreyer et al. 2003; Pant et al. 2004; Landis and Theis 2008). Table 1 shows that disagreement in the conclusion of LCA studies when employing different life cycle impact assessment (LCIA) methodologies is observed for many impact categories, including human toxicity and ecotoxicity, photochemical ozone formation, ozone depletion, acidification and eutrophication. This disagreement is a source of criticism of LCA (Finnveden et al. 2009) and has led many

Table 1 Reasons for disagreement and some agreement in conclusions of LCA studies when employing different LCIA methods

Reason	Impact category	References
Disagreement		
Differences in the underlying characterization model	Global warming	Landis and Theis (2008)
	Ecotoxicity	Dreyer et al. (2003); Pant et al. (2004); Bovea and Gallardo (2006); Pizzol et al. (2011a)
	Human toxicity	Dreyer et al. (2003); Bovea and Gallardo (2006); Landis and Theis (2008); Renou et al. (2008); van Caneghem et al. (2010); Pizzol et al. (2011b); Zhou et al. (2011)
	Photochemical ozone formation	Bovea and Gallardo (2006); Landis and Theis (2008); Cellura et al. (2011)
	Acidification	Landis and Theis (2008)
	Eutrophication	Landis and Theis (2008)
Differences in substance coverage	Global warming	de Alvarenga et al. (2012)
	Ecotoxicity	Dreyer et al. (2003); Bovea and Gallardo (2006); Pizzol et al. (2011a); Zhou et al. (2011)
	Human toxicity	Dreyer et al. (2003); Bovea and Gallardo (2006); Landis and Theis (2008); Pizzol et al. (2011b); Zhou et al. (2011)
	Photochemical ozone formation	Dreyer et al. (2003); Bovea and Gallardo (2006); Cellura et al. (2011); Zhou et al. (2011)
	Ozone depletion	Zhou et al. (2011)
Differences in relative ranking of the reference substance	Ecotoxicity	Dreyer et al. (2003)
	Human toxicity	Dreyer et al. (2003)
Differences in spatial scales and reference years for normalization references	Global warming	Dreyer et al. (2003)
	Ozone depletion	Dreyer et al. (2003)
	Acidification	Dreyer et al. (2003)
	Eutrophication	Dreyer et al. (2003)
	Photochemical ozone formation	Dreyer et al. (2003)
Combining similar compounds into a group with single characterization factor	Photochemical ozone formation	Zhou et al. (2011)
Different approaches to weighting	Human toxicity	Landis and Theis (2008)
Some agreement		
Similarities in the underlying characterization model	Global warming	Renou et al. (2008); Zhou et al. (2011)
	Eutrophication	Dreyer et al. (2003)
	Acidification	Cellura et al. (2011)
	Ozone depletion	Cellura et al. (2011)
Limited number of contributing substances	Ozone depletion	Dreyer et al. (2003); Cellura et al. (2011)
One or few processes driving impacts	Acidification	Monteiro and Freire (2012)
	Eutrophication	Monteiro and Freire (2012)
	Climate change	Monteiro and Freire (2012)
	Resource depletion	Renou et al. (2008); Monteiro and Freire (2012)
	Land use	de Alvarenga et al. (2012)
	Various single-score indicators	Huijbregts et al. (2010)
Differences compensating each other	Acidification	Zhou et al. (2011)

LCA practitioners to exclude some potentially important impact categories from assessments (e.g. Laurent et al. 2014).

Some of the recent advances in LCIA include the development of ReCiPe 2008 methodology (Goedkoop et al. 2009) and the development of the consensus characterization model USEtox (Hauschild et al. 2008; Rosenbaum et al. 2008). In 2012, following a comparison of existing characterization methods involving hearing of domain experts and stakeholders at large, the Joint Research Centre of the European Commission identified best practice and launched a recommended set of characterization models and factors for application in LCIA (EC-JRC 2012; Hauschild et al. 2013). The recommended method, further referred to as ILCD 2009, was compiled by assessing a total of 156 different characterization models belonging to 12 different LCIA methodologies and choosing the most appropriate, based on a predefined set of assessment criteria (EC-JCR 2011). It represents what was best practice in 2008–2009. The ILCD 2009 is now being introduced into LCA modelling tools, but it is not known yet whether there can be differences in impact scores between the ILCD 2009 and other frequently used LCIA methodologies, such as IMPACT 2002+ (Jolliet et al. 2003) or ReCiPe 2008 (Goedkoop et al. 2009), and whether the choice of the ILCD 2009 matters for the interpretation of LCA results.

The aim of our study was to investigate consequences of choosing different LCIA methods on the interpretation phase of an LCA. The ILCD 2009 is the focus of this paper, because its performance relative to IMPACT 2002+ or ReCiPe 2008 has not yet been studied. All three methodologies span a range of approaches to characterization modelling that have been proposed in the last decade (Hauschild et al. 2013). As a vehicle for the comparison, a case study comparing four window design options for use in a residential building was used. First, the influence of the LCIA method on ranking of the window options was investigated for each impact category within each methodology. Next, impact scores were converted into common metrics for each impact category to allow comparison across the three methodologies both in terms of total results and contribution patterns for individual substances. The focus is on characterization as no normalization and weighting factors are available for the ILCD 2009 methodology.

2 Methods

2.1 Selection of impact categories

All impact categories that are common to at least two of the three methodologies were included in the comparison (Table 2). Ionizing radiation impacts on ecosystems (included only in the ILCD 2009) and aquatic acidification (included only in IMPACT 2002+) were thus excluded. The life cycle inventory data on water use were of insufficient quality, so the water use impact category was also excluded. The list of compared impact categories is shown in Table 2. Note that six impact categories of the ILCD 2009 (climate change, stratospheric ozone depletion, photochemical ozone formation, freshwater eutrophication, marine eutrophication and impact from ionizing radiation to human health) are based on the models and factors applied in ReCiPe 2008. Furthermore, all studied methodologies utilize different versions of the climate change model developed by the Intergovernmental Panel on Climate Change (IPCC) (Forster et al. 2007). Only characterization at midpoint was compared as the ILCD provides few recommendations at the endpoint level due to the immaturity of the existing approaches (Hauschild et al. 2013).

2.2 Conversion of impact scores into common metrics

To compare impact scores across methodologies, the impact scores for each impact category were converted into common metrics, following the approach proposed by Dreyer et al. (2003) (Eq. 1).

$$\mathrm{IS}_j = \mathrm{IS}_i \times \mathrm{UCF}_{i\rightarrow j} \tag{1}$$

where IS_j is the impact score expressed in a unit of a new reference substance for a given impact category, IS_i is the impact score in old units for that impact category, and $\mathrm{UCF}_{i\rightarrow j}$ is the method- and impact category-specific unit conversion factor, defined as the reciprocal of a characterization factor for the new reference substance (Eq. 2).

$$\mathrm{UCF}_{i\rightarrow j} = \frac{1}{\mathrm{CF}_i} \tag{2}$$

where CF_i is the characterization factor for the new reference substance, expressed in original units i.

The criteria for selection of a reference substance were the following: (1) it must have a CF in all compared methods, and (2) it must be a reference substance in at least one of the compared methods. Exceptions were made for impacts from terrestrial acidification and eutrophication, which are either combined into one impact category or split, depending on the methodology.

2.2.1 Unit conversion factors

Unit conversion factors are show in Table 2. For climate change and stratospheric ozone depletion, the metric is the same across all methods and impact scores can be compared directly. For other impact categories, the units had to be converted. Sulphur dioxide (SO_2) is used as reference substance in IMPACT 2002+ for the combined effects on

Table 2 Midpoint impact categories and unit conversion factors ($UCF_{i \rightarrow j}$) between LCIA methods included in ILCD 2009, ReCiPe 2008 and IMPACT 2002+

Methodology and impact category [unit i]			Unit conversion factor, $UCF_{i \rightarrow j}$ [a]			Reference unit *j*
ILCD 2009	ReCiPe 2008	IMPACT 2002+	ILCD 2009	ReCiPe 2008	IMPACT 2002+	
Climate change [kg CO_2 eq]*		Global warming 500yr [kg CO_2 eq to air]	1	1	1	kg CO_2 eq
Ozone depletion [kg CFC-11 eq]*		Ozone layer depletion [kg CFC-11 eq to air]	1	1	1	kg CFC-11eq
Photochemical ozone formation [kg NMVOC eq]*		Photochemical oxidation [kg C_2H_4 eq to air]	1	1	1.66	kg NMVOC eq
Acidification [AE]	Terrestrial acidification [kg SO_2 eq]	Terrestrial acidification/ nutrification [kg SO_2 eq to air]	3.31×10^{-1}	4.08×10^{-1}	6.69×10^{2}	kg NH_3 eq
Eutrophication, terrestrial [AE]	NA		7.42×10^{-2}	NA		kg NH_3 eq
Freshwater eutrophication [kg P eq]*		Aquatic eutrophication [kg PO_4^{3-} eq to water]	3.03	3.03	1	kg PO_4 eq
Marine eutrophication [kg N eq]*			4.24×10^{-1}	4.24×10^{-1}		kg PO_4 eq
Ecotoxicity, freshwater [CTU_e]	Ecotoxicity, freshwater [kg 1,4-DB eq]	Aquatic ecotoxicity [kg TEG eq to water]	1.02×10^{-3}	1	2.05×10^{-5}	kg 1,4 DB eq (freshwater)
NA	Ecotoxicity, marine [kg 1,4-DB eq]		NA	4.69		kg 1,4 DB eq (freshwater)
NA	Ecotoxicity, Terrestrial [kg 1,4-DB eq]	Terrestrial ecotoxicity [kg TEG eq to soil]	NA	1.97×10^{2}	2.46	kg 1,4 DB eq (freshwater)
Ionising radiation, human health [kBq U235 eq] *		Ionizing radiation [Bq C-14 eq to air]	1	1	1×10^{-1}	kBq U235 eq
Particulate matter/ respiratory inorganics [kg $PM_{2.5}$ eq to air]	Particulate matter formation [kg PM_{10} eq]	Respiratory effects – Midpoint [kg $PM_{2.5}$ eq to air]	1	1	1	kg $PM_{2.5}$ eq
Human toxicity, cancer effects [CTU_h]	Human toxicity [kg 1,4-DB eq]	Carcinogens [kg C_2H_3Cl eq to air]	4.65×10^{6}	1	5.41×10^{2}	kg 1,4-DB eq
Human toxicity, non-cancer effects [CTU_h]		Non-carcinogens [kg C_2H_3Cl eq to air]	1.56×10^{7}		1.74×10^{2}	kg 1,4-DB eq
Land use [kg C·yr]	Urban land occupation [m^2·yr]	Land occupation [m^2·yr-org. arable eq]	1.03×10^{-1}	1	1	m^2·yr-org. arable eq
	Agricultural land occupation [m^2·yr]			1		
	Natural land transformation [m^2]	NA		NA	NA	NA
Resource depletion (mineral, fossils) [kg Sb eq]	Metal depletion [kg Fe eq]	Mineral extraction [MJ surplus]	NA	1	1.96×10^{1}	kg Fe eq
	Fossil depletion [kg oil eq]	Non-renewable energy [MJ]	NA	1	2.39×10^{-2}	kg oil eq

[a] NA indicated that the unit conversion factor could not be calculated. *Stars* indicate that the ILCD 2009 models and factors are based on ReCiPe 2008. See Section S1 of the Electronic Supplementary Material for details on calculation of unit conversion factors and comparison across LCIA methods

terrestrial acidification and terrestrial eutrophication but does not contribute to the latter; this means that the comparison was only possible by selecting a substance common for all characterization models in both impact categories, namely ammonia (NH_3). For aquatic eutrophication, phosphate (PO_4^{3-}) was chosen because it is the reference substance in IMPACT 2002+ (where freshwater and marine eutrophication impacts are aggregated into one impact category). As in IMPACT 2002+, freshwater and marine eutrophication impacts in ILCD 2009 (being ReCiPe 2008) were aggregated using the Redfield conversion ratio between phosphate and nitrogen compounds (Goedkoop et al. 2009). For freshwater ecotoxicity, conversion from 1,4-dichlorobenzene equivalents (1,4-DB eq) or triethylene glycol equivalents (TEG eq) to the ILCD 2009 unit based on potentially affected fraction (PAF) of species ($m^3\ kg^{-1}$ day) was associated with larger uncertainties, as discussed in Section S1.6 of the Electronic Supplementary Material. Therefore, 1,4-DB was chosen as the reference substance. For the same reason, 1,4-DB was chosen for the impact categories representing human toxicity. For land use impacts on natural environment, comparison between the three methodologies was made possible by expressing scores in equivalents of arable land area occupied over time (that is, the unit in IMPACT 2002+). ReCiPe 2008 defines its midpoint characterization model for land use assuming competitiveness of the land types (where all CFs are equal to 1). Hence, the unit conversion factors for ReCiPe 2008 land occupation impact categories are equal to 1. Impacts from land transformation are expressed in different units (that is, land area) and were not considered. Note that the ILCD 2009 covers impacts from both land occupation and transformation, whereas both ReCiPe 2008 and IMPACT 2002+ only cover impacts from land occupation. For resource depletion, comparisons between ILCD 2009 (where minerals and fossils are combined) and either IMPACT 2002+ or ReCiPe 2008 were not possible because combining mineral and fossil depletion indicators in the two latter methods was too uncertain (see Section S1.11 of the Electronic Supplementary Material). Therefore, we only compared impact scores for resource depletion between IMPACT 2002+ and ReCiPe 2008. Details of the unit conversion are documented in Section S1 of the Electronic Supplementary Material.

2.3 Identifying reasons for differences in impact scores

Differences in impact scores (converted into common metrics) between LCIA methods can be due to (i) differences in underlying characterization models, (ii) differences in substance coverage and/or (iii) errors in implementation of characterization factors into the modelling software. We quantified the contribution of each of these causes to the total difference in impact scores between the studied methods for wood window system. First, most contributing substances were identified, and their contribution to total impact score was quantified (see Section S3 of the Electronic Supplementary Material for details on substance contribution analysis). For many impact categories, the number of environmental flows was large, so flows contributing to <1 % of the total impact were combined into category “others”. This is not expected to influence our result because contribution from this category was below 3 % of the total impact score across all impact categories. The difference in impact scores between those substances which are common for the two LCIA methods reflects cause (i) (that is, difference due to underlying characterization models). To assess the differences in impact scores due to substance coverage, substances contributing to impact score in one LCIA method but not in another were separately investigated with regard to their presence in the original characterization factor data sets (reflecting cause (ii)) and in the set of CFs actually implemented in the LCA software (reflecting cause (iii)). The contribution of the three causes to the total difference in impact scores was quantified using Eq. 3.

$$\Delta \mathrm{IS}^{(i),(ii),\mathrm{or}\ (iii)}(\%) = \frac{\mathrm{IS}^{(i),(ii),\mathrm{or}\ (iii)}_{\mathrm{compared\ method}} - \mathrm{IS}^{(i),(ii),\mathrm{or}\ (iii)}_{\mathrm{reference\ method}}}{\mathrm{IS}^{\mathrm{total}}_{\mathrm{compared\ method}} - \mathrm{IS}^{\mathrm{total}}_{\mathrm{reference\ method}}} \times 100\% \tag{3}$$

where $\Delta \mathrm{IS}^{(i),\ (ii),\mathrm{or}\ (iii)}$ (%) is the contribution from (i) differences in characterization models, (ii) substance coverage or (iii) error in implementation of characterization factors into modelling software to the total difference in impact score between the compared and reference methods; $\mathrm{IS}^{(i),(ii),\mathrm{or}\ (iii)}_{\mathrm{compared\ method}}$ and $IS^{(i),(ii),\mathrm{or}\ (iii)}_{\mathrm{reference\ method}}$ are the impact scores for the contributing substances in cases (i), (ii) or (iii) for the compared and the reference methods, respectively, and $\mathrm{IS}^{\mathrm{total}}_{\mathrm{compared\ method}}$ and $\mathrm{IS}^{\mathrm{total}}_{\mathrm{reference\ method}}$ are the total impact scores for the compared and the reference methods, respectively.

Note that the differences in impact scores due to any of the considered causes can be larger than the difference in total impact scores (resulting in contribution from that cause of >100 %). In such situations, it means that there has to be a negative contribution from one (or two) remaining cause that compensates for this difference, because the sum of contributions from the three causes has to be equal to 100 %. Section S4 of the Electronic Supplementary Material illustrates this for the climate change and photochemical ozone formation impact categories, chosen as examples.

3 Case study

A comparative cradle-to-grave LCA for four window alternatives used in a residential building in Denmark was conducted

in accordance with the requirements of the ISO standard and the ILCD Handbook (ISO 2006; EC-JRC 2010). Details on the case study are presented in Section S2 of the Electronic Supplementary Material.

3.1 Goal definition

Windows can account for 25 % of heat loss in a residential building, and one way to reduce heat loss is to introduce products with better insulation properties (Natural Resources Canada, 2012). Focus in the product development is therefore natural on improving the insulating properties of windows throughout their use stage. To optimize the environmental performance of the windows, it is important to evaluate the trade-off between expected lower impacts due to better insulation properties and negative environmental impacts from the need to use alternative materials or processes in the manufacturing of the window. The goal of the study is to find the window design with the lowest environmental impact in a life cycle perspective, taking into account the heat loss through the window during the use stage.

3.2 Scope definition

The functional unit is "allowing daylight into a building equivalent to light being transmitted through an area of 1.23×1.48 m^2 with visible light transmittance of at least 0.7, for 20 years". The calculation of heat demand for the building while the functional unit is fulfilled is based on average outdoor and indoor temperatures of 7.7 and 17 °C, respectively.

The chosen windows are representative to windows currently offered in the market with respect to materials used and energy requirements (Energivinduer 2013). Major properties of each window type are shown in Table 3. The product systems include raw material extraction and production of materials. Primary data on energy and material consumption were collected from literature and were mapped to processes in the Ecoinvent database version 2.2 (Hischier et al. 2010) combined with the PlasticsEurope (PE) database (www.plasticseurope.org). Data inventory for background and specific processes are based on average technology. Data sources, bills of materials and energy, product systems and unit processes are documented in Section S2 of the Electronic Supplementary Material.

The decision context in terms of the ILCD guideline is scope situation A (microlevel decision support), suggesting that the attributional principle be chosen as the LCI modelling framework. This implies that the systems are modelled depicting existing value chains, that is, using current Danish electricity and heating mix in production and use stage, and Danish end-of-life scenarios. We moreover assumed that the windows are used only in buildings to which heat is provided by district heating. In Denmark, district heating delivered ~50 % of the total heat demand for buildings in 2010 and is based on biomass (18 % straw and 16 % wood), natural gas (34 %), coal (24 %), non-renewable waste (6 %) and oil (3 %) (Dyrelund and Lund 2008; Danish Energy Agency 2011). Heat production was modelled using Norwegian and Swiss technology (see Table S10, Electronic Supplementary Material). We tested the relevance of these assumptions by comparing the impact scores between window options using average European technology and heat mix, which are based on natural gas (57 %), oil (21 %), biomass (13 %) and coal (9 %) (EC 2012). For the recycling of materials, system expansion has been performed, with credits given for avoided primary production of virgin materials (such as aluminium ingots, steel billet, glass cullets or the PVC granulate mix) and to incineration for avoided production of heat and electricity (using Danish mixes). The three wood-based windows are sold mainly in Sweden and Denmark, but the use and disposal stages for all windows are modelled as if occurring in Denmark. This also applies for the polyvinyl chloride (PVC)

Table 3 Major properties of the studied windows

Property	Window type			
	W	W/ALU	PVC	W/C
Frame design	Pine wood	Pine wood and aluminium	PVC, steel support	Pine wood and glass fibre polyamide
Pane design	Two-layered, sealed with silicone, filled with argon	Two-layered, sealed with silicone, filled with argon	Two-layered, sealed with silicone, filled with argon	Two-layered, sealed with silicone, filled with argon
Pane area (m^2)	1.82	1.82	1.82	1.82
Overall heat transfer coefficient (U value, in W m^{-2} K^{-1})	1.29	1.31	1.36	1.08
Visible light transmittance (T_{vis}, in fraction)	0.8	0.8	0.8	0.8

window, which is sold mainly in Western Europe. All windows are disposed of according to the Danish waste policy, where glass, aluminium and steel are mainly recycled, and wood is incinerated. Other materials are either incinerated or landfilled. PVC is technically recyclable, but its recycling rate in Denmark is lower compared to other plastics (30 %) (PlasticsEurope 2012). Several assumptions had to be made with respect to modelling individual processes and end-of-life scenarios. Chromium steel and galvanized steel were modelled using the same unit process, while landfilling of EDPM rubber was modelled as that of polypropylene. Further, assumptions had to be made for transportation means and distances. Details on the case study, including processes involved in the life cycle of all four window alternatives and the list emissions per functional unit, are shown in Section S2 of the Electronic Supplementary Material.

3.3 Characterization factors and system modelling

Product systems were modelled in GaBi, version 4.3 (PE International, Germany). The ILCD 2009 was not implemented into GaBi at the time of the study; thus, characterization factors for the ILCD methods (version 1.0.3, 01 March 2012) were downloaded from the Life Cycle website of the European Commission (http://lct.jrc.ec.europa.eu/assessment) and were imported into the software. For those ILCD 2009 methods where ReCiPe 2008 is the recommended method, impact scores were calculated using the original set of ReCiPe (version 1.05) characterization factors as implemented in GaBi. The same set of ReCiPe (version 1.05) characterization factors, and a set of IMPACT 2002+ (version 2.1) characterization factors implemented in GaBi, were employed to calculate impact scores for ReCiPe 2008 and IMPACT 2002+, respectively. For freshwater ecotoxicity, impact scores were calculated off-line using emission inventory data exported from GaBi and characterization factors from USEtox, version 1.01 (Rosenbaum et al. 2008).

4 Results and discussion

The results below show the ranking of the four window options relative to the impact score for the PVC window and the comparisons between impact scores converted into common metrics. Reasons for differences in impact scores and implications of our findings for LCA of products are then discussed.

4.1 Agreement in ranking of window options

Figure 1 shows that the environmental impacts expressed relative to the scores for the PVC window are within the same order of magnitude for all impact categories. The ranking of window alternatives generally follows the order W/C>W≈W/ALU>PVC, except for toxicity- and land use-related impacts. The W/C window is shown to be the best for all of the ILCD 2009 impact categories, 17 out of 18 ReCiPe impact categories and 14 out of 15 IMPACT 2002+ impact categories. The PVC window is seen as the worst alternative in 10 out of 15 ILCD 2009 impact categories, 14 out of 18 ReCiPe 2008 impact categories and 13 out of 15 IMPACT 2002+ impact categories. The ranking between the two other alternatives (W and W/ALU) varies, depending on the methodology and impact category.

For non-toxic impacts, all three methodologies generally agree on the best (W/C) and the worst (PVC) alternative (note that for most non-toxic impact categories, ReCiPe 2008 is the ILCD-recommended method). However, the W window is consistently seen as the worst alternative with all three methodologies for ozone depletion. For toxicity-related impacts, ILCD 2009 disagrees with ReCiPe 2008 and IMPACT 2002+ in ranking for human toxicity (non-cancer effects) and freshwater ecotoxicity. For the latter, the W window is seen as the worst in ILCD 2009, while the PVC window is the worst according to both ReCiPe 2008 and IMPACT 2002+. For land use, all three methodologies generally agree in determining the worst alternative, namely the W window. Land occupation provides a similar profile in both ReCiPe 2008 and IMPACT 2002+, yet this is based on the assumptions that agricultural land occupation has larger contribution than urban land occupation. For land use in ILCD 2009, the ranking is similar to that for natural land transformation in ReCiPe 2008, suggesting that impacts from land use primarily stem from land transformation. Similar ranking is obtained across methodologies for metal/mineral depletion and fossils depletion, with the PVC window shown to be the worst and W/C window system the best. Even though the ILCD 2009 methodology aggregates scores for metal/mineral and fossil depletion, the ranking is very similar to that for IMPACT 2002+ and ReCiPe 2008.

The apparent agreement between the studied methodologies in ranking of window alternatives contrasts with many reports of the dependence of LCA results on the chosen methodology (see Table 1). However, our findings are in agreement with studies documenting that different impact assessment methods are expected to provide converging results if one process is the main driver of environmental impact (Huijbregts et al. 2010). In the window study, the main driver of environmental impacts is the production of household heating to compensate for heat losses through the window (see Section S5 of the Electronic Supplementary Material for details on the contribution of the use stage to total impact

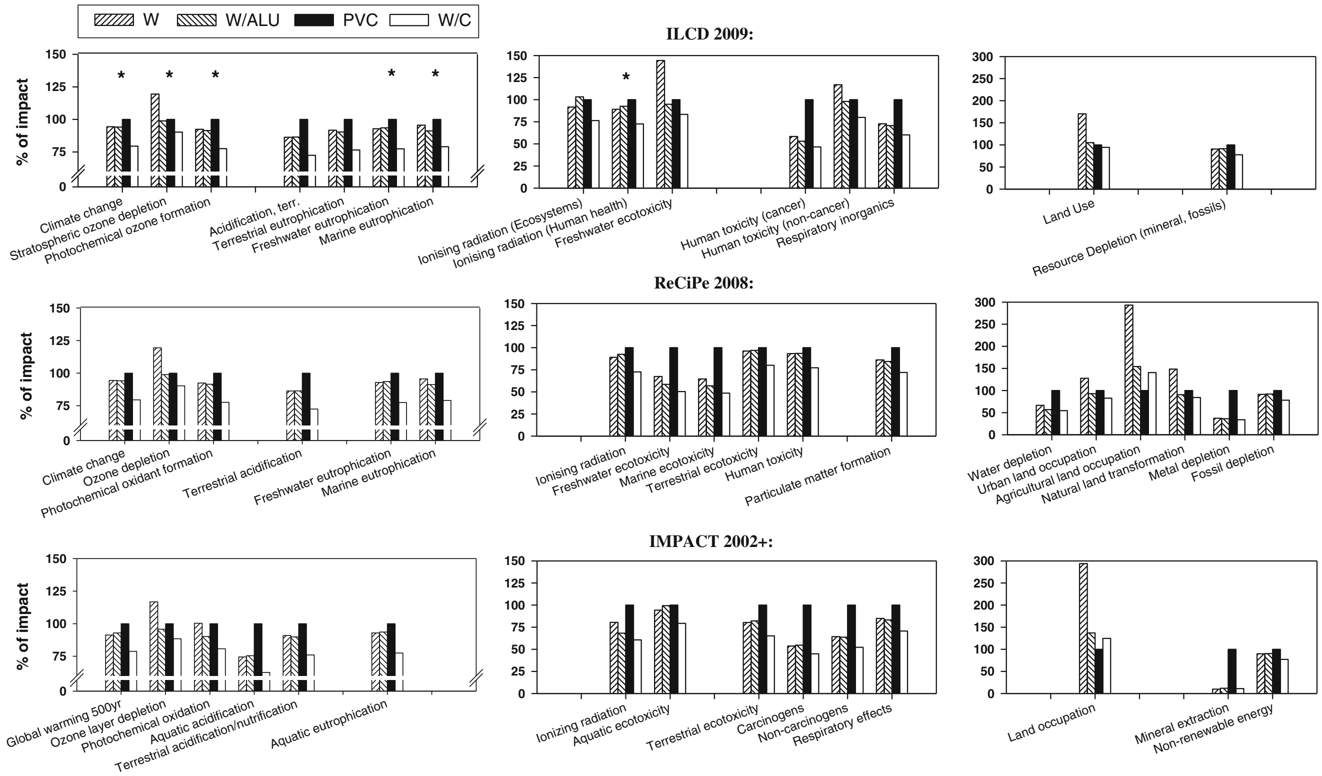

Fig. 1 Ranking of four window options in ILCD 2009 (*top*), ReCiPe 2008 (*middle*) and IMPACT 2002+ (*bottom*). Impact scores are scaled to those of the PVC window, set equal to 100 %. Impact categories for the methodologies are named as provided in the original documentation. *Stars* indicate that the ILCD 2009 models and factors are based on ReCiPe 2008. Results for all impact categories available within considered methodology are shown

scores). The differences in demand for heat between the best (W/C) and the worst (PVC) alternatives are large enough to allow the different impact assessment methods to agree in product ranking. By contrast, the small difference in demand for heat between the W and W/ALU alternatives results in environmental impacts, which are so similar, that product ranking depends largely on the method chosen. Small differences in ranking between W and W/ALU window options suggest that if the uncertainties associated with the input data and characterization models were considered, none of the methods would be expected to give a straightforward answer on the second best/worst alternative (Huijbregts et al. 2003).

To see whether the choice of Danish heat mix may influence our conclusions, we carried out a sensitivity analysis of the use stage using the European average heat mix. For the average European heat mix, the W/C window is shown to be the best for all of the ILCD 2009 impact categories, 17 out of 18 ReCiPe impact categories and 13 out of 15 IMPACT 2002+ impact categories. By contrast, the PVC window is seen as the worst alternative in 12 out of 15 ILCD 2009 impact categories, 16 out of 18 ReCiPe 2008 impact categories and 13 out of 15 IMPACT 2002+ impact categories (data now shown). This suggests that the ranking of window options according to the three methodologies is robust to the heat mix chosen.

4.2 Category-specific differences in impact scores

Figure 2 shows that the impact scores converted into common metrics differ across methodologies. Impact scores are within a factor of 3 for the impact categories climate change, stratospheric ozone depletion, acidification and terrestrial eutrophication, aquatic eutrophication, freshwater ecotoxicity (between ILCD and ReCiPe 2008) and fossil depletion. Impact scores within 1 order of magnitude are observed for the categories respiratory inorganics and photochemical ozone formation. For aquatic ecotoxicity (aggregated over marine water and freshwater), metal depletion and toxic impacts on human health, impact scores show a difference from 1 to 3 orders of magnitude. The differences in impact from ionizing radiation on human health and impacts from land use are above 3 orders of magnitude.

Table 4 shows that both differences in the underlying characterization model and substance coverage can bring about a difference in impact scores, depending on the impact category. It also shows that there are occasional errors in the implementation of CFs for IMPACT 2002+ or ReCiPe 2008 in the applied modelling software. The results presented in Fig. 2 and Table 4 for the wood (W) window option, accompanied by substance contribution analysis presented in

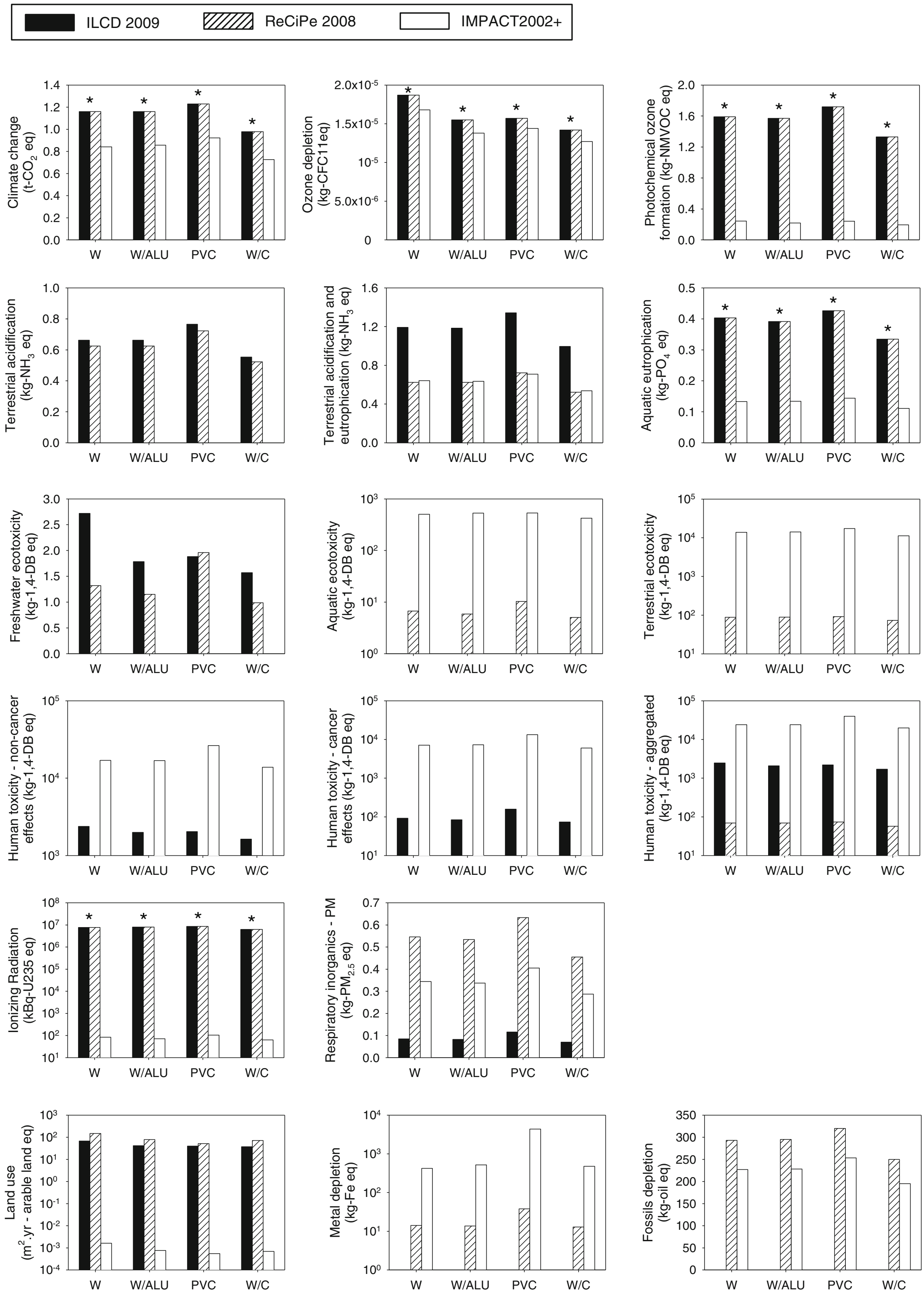

Fig. 2 Impact scores converted into common metrics for impact categories listed in Table 2. *Stars* indicate that the ILCD 2009 models and factors are based on ReCiPe 2008

Table 4 Contribution of differences in characterization models, substance coverage and software implementation to differences in impact scores between ILCD 2009, IMPACT 2002+ and ReCiPe 2008 for the wood (W) window system

IMPACT category	Reference LCIA method	Compared LCIA method 1	Cause of discrepancies and contribution (%)			Compared LCIA method 2	Cause of discrepancies and contribution (%)		
			Characterization model	Substance coverage[a]	Software implementation		Characterization model	Substance coverage[a]	Software implementation
Climate change	ILCD 2009	–	–	–	–	IMPACT 2002+	15	85	0
Ozone depletion	ILCD 2009	–	–	–	–	IMPACT 2002+	−1	101	0
Photochemical ozone formation	ILCD 2009	–	–	–	–	IMPACT 2002+	3	113	−16
Terrestrial acidification and eutrophication	ILCD 2009	ReCiPe 2008	100	0	0	IMPACT 2002+	100	0	0
Terrestrial acidification	ILCD 2009	ReCiPe 2008	100	0	0	–	–	–	–
Aquatic eutrophication	ILCD 2009	–	–	–	–	IMPACT 2002+	100	0	0
Freshwater ecotoxicity	ILCD 2009	ReCiPe 2008	113	−13	0	–	–	–	–
Aquatic ecotoxicity	ReCiPe 2008	–	–	–	–	IMPACT 2002+	0	99	1
Terrestrial ecotoxicity	ReCiPe 2008	–	–	–	–	IMPACT 2002+	6	−1	94
Human toxicity, aggregated effects	ILCD 2009	ReCiPe 2008	65	35	0	IMPACT 2002+	89	11	0
Human toxicity, cancer effects	ILCD 2009	–	–	–	–	IMPACT 2002+	99	−1	
Human toxicity, non-cancer effects	ILCD 2009	–	–	–	–	IMPACT 2002+	15	84	1
Ionizing radiation (human health)	ILCD 2009	–	–	–	–	IMPACT 2002+	100	0	0
Respiratory inorganics	ILCD 2009	ReCiPe 2008	95	5	0	IMPACT 2002+	86	14	0
Land use	ILCD 2009	–	–	–	–	IMPACT 2002+	54	46	0
Metal depletion	ReCiPe 2008	–	–	–	–	IMPACT 2002+	99	1	0
Fossil depletion	ReCiPe 2008	–	–	–	–	IMPACT 2002+	108	−8	0

[a] Situations, in which uncovered substances in both LCIA methods compensate each other, may occur although it is not reflected by the distributions reported in this column. Those details can however be viewed in Section S3 of the Electronic Supplementary Material

Section 3 of the Electronic Supplementary Material, are discussed per impact category below.

4.2.1 Climate change

Abiotic and biotic carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) contribute to more than 99.5 % of the total impacts for climate change in our product systems for both methods. A total difference in impact scores of 28 % between ILCD 2009 (being ReCiPe 2008) and IMPACT 2002+ originates from the fact that temporary carbon storage in the products is only credited in the latter. This difference, classified as a difference in substance coverage, accounts for 85 % of the total difference. The remaining 15 % can be attributed to differences in approaches to characterization modelling: a 500-year time horizon is considered in IMPACT 2002+, which corresponds to lower CFs for CH_4 and N_2O, compared to a 100-year time horizon in ILCD 2009. In 2012, IMPACT 2002+ developers adapted the climate change model for a 100-year time horizon, but this change was not yet implemented in the software.

4.2.2 Ozone depletion

ILCD 2009 (being ReCiPe 2008) and IMPACT 2002+ use different versions of a stratospheric ozone depletion model developed by the World Meteorological Institute (WMO) (Struijs et al. 2009). In our product systems, bromotrifluoromethane (halon 1301), dichlorodifluoromethane (R12—now abandoned in accordance with the Montreal Protocol) and bromochlorodifluoromethane (halon 1211) are the major (>75 %) contributors to impacts from ozone depletion. The difference of 10 % between ILCD 2009 (being ReCiPe 2008) and IMPACT 2002+ is due to a difference in substance coverage, as no CF is attributed to substances grouped into category "hydrocarbons, chlorinated" in the latter.

4.2.3 Photochemical ozone formation

ILCD 2009 (being ReCiPe 2008) applies the LOTOS-EUROS model to characterize the impact of photochemical ozone formation on human health (van Zelm et al. 2008). In contrast, midpoint characterization factors in IMPACT 2002+ are derived by dividing the damage factor taken directly from Eco-indicator 99 (Goedkoop and Spriensma 2000) by the damage factor of the reference substance (ethylene, C_2H_4). Eco-indicator 99 was not included in the comparison of LCIA methodologies because it had been superseded by the ReCiPe 2008 method (Hauschild et al. 2013). Despite these apparently distinct approaches to characterization modelling, the observed factor of 6 between impact scores across methods is mainly explained by differences in substance coverage (113 % of the total difference), as nitrogen oxides (NO_x) which account for 95 % of the total impact in ILCD 2009 (being ReCiPe 2008) do not have CFs in IMPACT 2002+. By contrast, in IMPACT 2002+, the major contributing substances (83 % of the total impacts) are non-methane volatile organic compounds (NMVOC). NMVOC are recognized as important contributors to the impacts of photochemical ozone formation in ILCD 2009 (being ReCiPe 2008), but they do not have a CF assigned in the employed modelling software. This is an error in software implementation, because the most important contributors to photochemical ozone formation in LOTOS-EUROS are NMVOC and NO_x. This error accounts for 16 % of the total observed difference in impact scores between the two methodologies.

4.2.4 Acidification, terrestrial

The most common substances recognized as causing acidification impacts in terrestrial ecosystems, ammonia (NH_3), nitrogen oxides (NO_x) and sulphur dioxide (SO_2), are present in our emission inventories. There is some disagreement in impact scores between ILCD 2009 and ReCiPe 2008, which is entirely caused by different characterization models. In ILCD 2009, the acidifying potency of a substance is modelled as the accumulated exceedance above the critical load in sensitive areas, including some important freshwater bodies (Seppälä et al. 2006; Posch et al. 2008), whereas in ReCiPe 2008, acidification is modelled as base saturation for European forest vegetation (van Zelm et al. 2007). As discussed below, in IMPACT 2002+, terrestrial acidification is combined with terrestrial eutrophication.

4.2.5 Eutrophication, terrestrial

Terrestrial eutrophication is mainly caused by nitrogen oxides and/or phosphate. Again, the accumulated exceedance principle is applied in ILCD 2009 to characterize impacts from airborne eutrophying substances. Direct comparison with ReCiPe 2008 and IMPACT 2002+ is not possible because the former does not include impacts on terrestrial eutrophication, whereas the latter combines impacts on terrestrial acidification with impacts on terrestrial eutrophication. However, our comparison between the combined impact scores of terrestrial eutrophication and acidification shows that a nearly twofold difference between ILCD 2009 and IMPACT 2002+ is solely caused by a difference in characterization models. Midpoint CFs in IMPACT 2002+ are derived from the damage CFs taken directly from Eco-indicator 99 (Goedkoop and Spriensma 2000), where characterization modelling is done without considering the atmospheric fate of a substance.

4.2.6 Eutrophication, aquatic

A nearly fourfold difference between ILCD 2009 (being ReCiPe 2008) and IMPACT 2002+ in impact scores for aquatic eutrophication is due to differences in the characterization models. The former includes substance fate and makes a distinction between limiting nutrients in aquatic bodies (P in freshwater and N in marine water), whereas midpoint CFs in IMPACT 2002+ are those from the CML 2002 methodology, where both substance fate and distinction between limiting nutrients are ignored (Guinée et al. 2002). The two methods also differ in substance coverage, but some of the substances (such as those grouped into chemical oxygen demand, COD) are not present in our life cycle inventory.

4.2.7 Ecotoxicity

For freshwater ecotoxicity, our emission inventory includes a total of 190 substances which are expected to cause toxic impact in freshwater ecosystems using ILCD 2009. The difference in impact scores between the ILCD 2009 (where USEtox is the recommended method) and ReCiPe 2008 is shown to be up to a factor of 2. The differences mainly originate from discrepancies in approaches to characterization modelling. The ILCD 2009 and ReCiPe 2008 agree that toxic impacts are dominated by emissions of metals, but the top-ranked elements differ between the two methods. Zinc(II) and copper(II) are responsible for 79 % of the total impact in ILCD 2009, whereas, in ReCiPe 2008, 82 % of the total impact is caused by emissions of nickel(II), zinc(II), bromine and cobalt. These observations are in agreement with conclusions from Pizzol et al. (2011a, b), who demonstrated how the choice of impact assessment method affects toxic impact scores for metals.

For marine ecotoxicity, no method is yet recommended in ILCD 2009. Our comparison between ReCiPe 2008 and IMPACT 2002+ for combined impacts on freshwater and marine ecosystems shows a difference of more than 1 order of magnitude. It appears that nearly all the discrepancy is attributed to differences in substance coverage, as toxic impacts in IMPACT 2002+ are dominated by aluminium(III), which does not have a CF in ReCiPe 2008. In addition, each of the substances that contribute to more than 1 % of the total impact in ReCiPe 2008 either was found to contribute less than 1 % of the total impact in IMPACT 2002+ or was not covered at all. This suggests that the difference in characterization models between ReCiPe 2008 and IMPACT 2002+ is also large. This difference is not apparent in Table 4 because the overall discrepancy is caused by not including aluminium(III) in ReCiPe 2008.

For terrestrial ecotoxicity, again, no method is yet recommended in ILCD 2009. Our comparison between ReCiPe 2008 and IMPACT 2002+ shows that the discrepancy in substance coverage is high; no single substance is the same among those which contribute to 99 % of total impacts. In ReCiPe 2008, impacts are dominated by phosphorus, whereas IMPACT 2002+ gives strong weight to impacts from metal emissions. Most of these metals do have a characterization factor in ReCiPe 2008, but they have not been implemented correctly in the software.

4.2.8 Human toxicity, cancer effects

A difference of about 2 orders of magnitude between ILCD 2009 (where again, USEtox is the recommended method) and IMPACT 2002+ can mainly be explained by the differences between characterization models. Polychlorinated dibenzo-*p*-dioxins (2,3,7,8-TCDD) are shown to be the top contributor in IMPACT 2002+, whereas they contribute to 1 % of the total impact in ILCD 2009. On the other hand, cancer effects in ILCD 2009 are dominated by chromium(VI) emitted to freshwater, but this emission route is not covered in IMPACT 2002+. There is rough agreement between the methods with respect to impact from arsenic(V), which contributes 3 and 14 % of the total impacts in ILCD 2009 and IMPACT 2002+, respectively (see Section S3 in Electronic Supplementary Material). As discussed below, ReCiPe 2008 aggregates impacts from carcinogens and non-carcinogens.

4.2.9 Human toxicity, non-cancer effects

A difference of about 1 order of magnitude between ILCD 2009 (where again, USEtox is the recommended method) and IMPACT 2002+ is mainly due to different substance coverage. This is mainly because 2,3,7,8-TCDD and aluminium(III), which are major contributors to non-cancer effects in IMPACT 2002+, are not included in ILCD 2009. Moreover, differences can also be attributed to differences in the characterization models; arsenic(V) and zinc(II) emitted to freshwater dominate impacts in ILCD 2009, but their contribution is lower in IMPACT 2002+.

Comparison between ILCD 2009 and ReCiPe 2008 for the combined cancer and non-cancer effects shows that emissions of zinc(II) and other substances which dominate impacts in ILCD 2009 are either less important or not included at all in ReCiPe 2008. The absolute contributions of zinc(II) in IMPACT 2002+ are comparable to those in ILCD 2009.

4.2.10 Ionizing radiation, human health

A difference of nearly 5 orders of magnitude between ILCD 2009 and IMPACT 2002+ in human health impacts from ionizing radiation is mainly caused by a difference in characterization models. Out of four substances that cause 99 % of the impact, three are covered by all the considered methods (^{14}C, ^{137}Cs and ^{222}Ra). However, the contribution of

individual substances differs between the methods. For our case study, the largest contributor is ^{14}C in ILCD 2009, while ^{230}Th and ^{222}Ra are the largest contributors in IMPACT 2002+. It should be noted that the latter is listed in ILCD 2009 as a contributor but does not have a CF implemented into the software.

4.2.11 Particulate matter/respiratory inorganics

Differences within 1 order of magnitude are observed between ILCD 2009, ReCiPe 2008 and IMPACT 2002+. They primarily originate from differences in the characterization models. Impacts in the two latter are dominated by NO_x, which is not seen as an important contributor in the former. By contrast, dust ($PM_{2.5}$) is seen as the largest contributor to this impact in ILCD 2009 (79 % of total impact) but does not exceed 25 % of the total impact in either of the two other methods.

4.2.12 Land use

A difference of nearly 6 orders of magnitude between ILCD 2009 and IMPACT 2002+ can be explained by the fact that these two models have fundamental differences in approaches to modelling land use impacts on natural environment. ILCD 2009 employs the method of Milà i Canals et al. (2007), where the impact indicator is based on soil organic matter (SOM), whereas IMPACT 2002+ uses indicators based on biodiversity, adapted from Eco-indicator 99 to a midpoint and excluding potential impacts from land transformation. In ReCiPe 2008, both land occupation (agricultural and urban) and land transformation have equal importance, assuming competitiveness among the land types.

4.2.13 Resource depletion, mineral and fossil

A comparison between ReCiPe 2008 and IMPACT 2002+ shows that a difference above 1 order of magnitude in metal depletion is mainly due to differences in characterization models. About 99 % of the impacts in ReCiPe 2008 are due to the depletion of 10 resources (with tin being the main contributor), whereas only five minerals/metals contribute to impact scores for IMPACT 2002+ (where nickel is the main contributor). Midpoint characterization factors in IMPACT 2002+ are derived from the damage-oriented Eco-indicator 99, which accounts for the surplus energy needed to exploit lower quality ore as a result of a marginal decrease in ore quality, while ReCiPe 2008 focuses on metal deposits, relying on the marginal increase of extraction costs. For fossil resource depletion, both ReCiPe 2008 and IMPACT 2002+ apply a method based on the calorific value, which results in comparable results both in terms of impact scores and relative contribution of substances. Aggregated impact scores for metal/mineral and fossil depletion in ILCD 2009 (being the CML 2002 method, Guinée et al. 2002) are very close to the impact scores in ReCiPe 2008 and IMPACT 2002+ for fossil depletion, suggesting that the depletion of fossils is the major contributor to the aggregated impact scores in ILCD 2009.

4.3 Limitations

We showed that the reasons for disagreement in impact scores for the wood (W) window are caused by the differences in underlying characterization models and/or substance coverage, depending on the impact category. Disagreement in impact scores can thus be expected for other product systems. It is important to stress, however, that no conclusions can be drawn with respect to the relative importance of differences in characterization models and differences in substance contribution for other product systems. These will vary from case to case, depending on the substances included in their emission inventories. For example, even if there are small differences in characterization models, impact scores can be sensitive to the inclusion of one or few substances with high characterization factors. This particularly holds true for impact categories where huge differences in characterization factors are possible, like the toxic impact categories where up to 12 orders of magnitude between characterization factors are observed (Rosenbaum et al. 2008). Indeed, the difference in impact scores across LCIA methods for the impact category freshwater ecotoxicity is smaller for the PVC window as compared with the W window, because different substances are present in emission inventories (Fig. 2 and Table S10, Electronic Supplementary Material).

5 Conclusions

Our case study presents a typical situation for an LCA practitioner, who has performed an inventory analysis and needs to make a decision about which LCIA methodology to use in order to answer the question posed in the goal definition (e.g. “which alternative has the lowest environmental impact?”). In cases like the one presented in this paper, where one or few processes dominate the total impacts for all compared options, and there is a large difference in demand for output from these processes between the compared options, it is likely that the methods will rank the compared products similarly. We thus expect that different LCIA methodologies will agree in ranking for those products, for which differences in demand for heat or electricity in the use stage are large. It is a challenge to the practitioner to identify whether this is the case.

Our study shows that after conversion to a common metric, discrepancies in impact scores between IMPACT 2002+, ReCiPe 2008 and the ILCD’s recommended practice can be large for the categories related to toxic impacts, ionizing

radiation, land use and mineral/metal depletion. This is supported by the contribution analyses showing substantial differences in contribution patterns between the four methodologies. We therefore recommend evaluating the ILCD's recommended practice for a wide range of goods and services. If discrepancies of the same order of magnitude are observed for other product systems for these impact categories, the practitioner should still employ the ILCD methods as the recommended practice, keeping in mind however that even the recommended practice either needs some improvements or is to be applied with caution for many impact categories (Hauschild et al. 2013). For the impact categories climate change, stratospheric ozone depletion, acidification and terrestrial eutrophication, aquatic eutrophication, fossil depletion, respiratory inorganics and photochemical ozone formation, the differences in impact scores across the studied methodologies are relatively small, and the choice of a methodology is not expected to influence the LCA results to the extent observed for other impact categories. Nevertheless, as a side remark for all three methodologies, attention should be paid to how characterization factors had been implemented into the modelling software employed.

In this study, we have only looked at the characterization step, but the normalization phase with its differences in normalization references between the methodologies is an extra potential source of difference, as could also be the weighting step (Dreyer et al. 2003). Modelling of impacts at endpoint is hardly supported by the ILCD's methods, but for the other methods, the applied approaches differ so widely that large discrepancies between the methodologies must be expected.

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